Overview of Chapter 7

7.0 Introduction

The Risk Integrated System of Closure (RISC) Technical Guide focuses almost exclusively on the default approach to risk-based closure. Default is defined as *the use of any constant, equation, model, process, strategy, or evaluation as identified within the RISC Technical Guide.*

The default approach represents a standard that IDEM will usually accept, except in those circumstances where the use of a particular default is inappropriate (for example, a default plume stability demonstration in a karst environment). Using the default approach can save time and transaction costs because generally the methods and values involved will require less extensive documentation and justification. The default approach may include the standard process for any of the following:

- Area screening
- Plume stability evaluation
- Closure sampling
- Default closure level
- Other standard procedures or inputs

As a non-rule policy RISC does not have the effect of law or rule; however, the default values and approaches have a sound technical basis and are considered valid approaches when applied to a broad range of scenarios encountered at remediation sites. The reader may view the default procedures described in this document as the methods preferred by IDEM except where such procedures have basis in rule or statute and are therefore requirements. The default closure process attempts to be a "one size fits all" approach. The simplest way to think of the nondefault approach is that it includes any pertinent procedure with a valid technical or policy basis that is not listed as a default IDEM preference. As a result, the limitations on the nondefault closure process are subject to interpretation regarding what is valid from a technical and policy perspective, and the nondefault process must be negotiated with the appropriate authority.

IDEM acknowledges that the default approach may not fit all situations and has developed the nondefault approach to provide a much greater degree of flexibility. This chapter provides a framework for using nondefault approaches within RISC.

Nondefault refers to *the use of any constant, equation, model, process, strategy, or evaluation that is not prescribed in the RISC Technical Guide for general application as a standard.* The nondefault approach is neither superior nor inferior to the default process. However, there are many reasons to consider nondefault applications, including accuracy, cost, necessity, and flexibility.

If a nondefault approach is employed, there will be a greater need to interact with IDEM technical review staff throughout the closure process. For example, a rationale for the technical validity of the nondefault application may be required (such as the technical rationale for sampling differently from the default approach while demonstrating that closure objectives have been obtained). The nondefault approach may also involve little more than relatively simple changes where both default and nondefault procedures are incorporated within a submittal. Examples of combined default and nondefault procedures include the following:

- Eliminating the migration to ground water pathway from further consideration for surface soil

- Substituting a smaller dilution attenuation factor in the default soil to ground water partitioning model when the subsurface soil source area exceeds ½ acre

- Using soil sampling results obtained during screening or characterization for a closure demonstration

Because of the greater uncertainty associated with the nondefault approach, IDEM recommends that such approaches be reviewed in a meeting with IDEM technical staff to explore options and identify expectations before submitting the risk assessment.

In some cases, the nondefault approach may be more desirable than a default approach because the nondefault approach may be more accurate on a site-specific basis. In the nondefault approach, site-specific information can be substituted for generic default information, (which is biased conservatively so it can be applied to a broad range of sites). The substitution of site-specific information may result in a higher closure level and subsequently, a less expensive closure. Nevertheless, the nondefault process may be more cost-effective at some sites. For example, site conditions may support collecting fewer samples than what is suggested for default. In other cases, a nondefault approach may be necessary because the default approach does not support the site conditions (for example, evaluating a source area greater than ½ acre for migration to ground water). A nondefault approach may also be necessary because the default approach does not consider an affected exposure pathway (such as ambient indoor air). For these and other reasons, nondefault procedures may be more applicable or advantageous at a particular site.

Potential disadvantages of using a nondefault approach should also be considered. Certain nondefault procedures will require greater technical sophistication on the part of the professional performing the evaluation (for example, probabilistic risk assessment). Other nondefault procedures may require more expensive technology (such as hydraulic conductivity testing for ground water modeling). Still other nondefault methods may require more specialized technical personnel (such as a toxicologist to evaluate dermal absorption factors, or a hydrogeologist to evaluate ground water modeling). The nondefault activity should be evaluated based on the value added by that activity. This outcome must balance protection of human health and the environment, cost, and public acceptance.

The guidance in this chapter covers general criteria that IDEM may use to evaluate a particular procedure as well as more specific detailed guidance on particular procedures.

7.1 Site-Specific Data That Can Be Used in the Default Equations

This section includes nondefault guidance for replacing default parameters in default equations with physical or chemical information that is specific to soil or ground water at a site.

A nondefault risk assessment allows closure levels to be derived using site-specific data in the soil-to-ground water partitioning model, the dilution attenuation factor (DAF) equation, the soil saturation limit equation, and the soil attenuation capacity equation. Site-specific data can also be used to determine the fraction of organic carbon and dry soil bulk density and soil porosity. Details on the use of site-specific values that can be used to replace default values are provided below.

7.1.1 Site-Specific Data for the Soil-to-Ground Water Partitioning Model

The soil-to-ground water partitioning model (see Equation 7-1 below) uses default ground water closure levels and site-specific soil data to calculate a migration to ground water soil constituent concentration. DAFs are used with the equation to account for natural constituent concentration reduction that occurs as constituents move through soil and ground water. Alternatively, a dilution factor may be substituted into the equation in place of the DAF. The following site-specific factors may be determined and used in Equation 7-1 to calculate a nondefault closure level for the migration to ground water pathway in soils:

- Fraction of organic carbon, determined specifically for surface or subsurface soil, whichever is appropriate

- Soil porosity and dry bulk density

- A site-specific dilution factor (see Equation 7-2) calculated using aquifer hydraulic conductivity, hydraulic gradient, infiltration rate, mixing zone depth, source length parallel to ground water flow, and aquifer thickness

The potential for constituents to migrate from soil to ground water prompts many cleanups; in such cases, reevaluating site-specific variables used in the soil-to-ground water partitioning model (also known as the migration to ground water model) may significantly affect closure levels. The model is based primarily on two principles:

- The constituent's equilibrium distribution between fractions sorbed to particles and fractions dissolved in solution (aqueous partitioning)

- The constituent's potential to migrate through the soil. Constituents that sorb tightly to soil organic matter are less likely to partition into the water phase within the soil pore space and are subsequently less likely to leach to ground water

The soil-to-ground water partitioning model estimates leachability using Equation 7-1. The model is most sensitive to the fraction of organic carbon (f_{oc}). If the site-specific value for f_{oc} is greater than the default value of 0.002 gram per gram (g/g), more of the constituent will remain sorbed to organic carbon in the soil, and less of the constituent will be available to leach to ground water. Changes to the other equation parameters are less likely to significantly affect closure levels for organic constituents.

Soil-to-Ground Water Partitioning Model

Equation 7-1. $$CL = C_w \times DAF \times \left[K_d + \frac{\theta_w + \theta_a H'}{\rho_b} \right]$$

Where:

CL = Closure level

C_w = Closure level for ground water (constituent specific in milligrams per liter)

DAF = Dilution attenuation factor (default value is equal to 20 for ½ acre, and 30 for 1/4 acre, or a site-specific DF may be substituted

DF = Dilution factor

K_d = Soil-water partition coefficient

For organic compounds, K_d *is equal to* K_{oc} *x* f_{oc} *where:*

K_{oc} = *Soil organic carbon-water partition coefficient (constituent specific in liters per kilogram)*

f_{oc} = *Organic carbon fraction of soil (default at 0.002 g/g)*

2_w = Water-filled soil porosity (default at 0.3 L water/L soil)

2_a = Air-filled soil porosity (default at 0.13 L air/L soil)

H' = Henry's Law Constant (dimensionless)

ρ_b = Dry soil bulk density (default at 1.5 kg/L)

K_d is the most significant factor in determining the leachability of metals. The *Soil Screening Guidance: User's Guide* (see Appendix C, Table C-4) lists certain metal K_d values as a function of pH. Default K_d values were selected assuming neutral soil pH (6.8). To calculate a site-specific closure level for the nondefault approach, K_d values may be adjusted for pH by substituting the values in the EPA's *Soil Screening Guidance, User's Guide*, Table C-4, for the default K_d in Equation 7-1. The pH evaluation should focus on soil within and immediately underlying the source area.

IDEM evaluated a study published by Sheppard and Thibault (1990) to determine an appropriate default K_d for lead migration to ground water. The study cautions that literature values are adequate for screening in simple systems, but such values should be used with caution, and preference should be given to site-specific information. In cases where the migration to ground water pathway is the limiting soil pathway for the migration of lead or other inorganics, IDEM suggests determining a site-specific leaching value using the Synthetic Precipitation Leaching Procedure (SPLP), EPA SW-846 Method 1312, EPA 1994d) or other appropriate analytical methods. IDEM recommends the following nondefault options for evaluating the migration to ground water pathway:

- Select a generic pH-specific K_d value (as referenced above)

- Use other leaching methods (such as SPLP) or other appropriate screen models that will measure or accurately predict site-specific leaching to ground water

7.1.2 Dilution Attenuation Factor

Both dilution and attenuation decrease the concentration of a constituent in ground water. Dilution occurs as the dissolved constituent disperses and mixes with less contaminated ground water. Attenuation occurs as the constituent is sorbed to soil or degrades through a variety of processes. To account for these processes, the soil-to-ground water partitioning model incorporates a DAF.

DAFs were selected based on the EPA *Soil Screening Guidance: Technical Background Document* (1996). DAFs represent conservative estimates of the dilution and attenuation that may occur at source areas of the sizes listed in Table 7-1 when the aquifer properties are homogeneous and isotropic.

Table 7-1. Dilution Attenuation Factors

Source Size	DAF
¼ acre or less	30
> ¼ acre to ½ acre	20
> ½ acre to 30 acres	10

Other DAFs may be proposed under a nondefault approach, provided adequate justification is given. For example, a particular source area may demonstrate a higher degree of dilution than is represented by the default DAFs. Equation 7-2 should be used to calculate a site-specific dilution factor which may then be substituted for the DAF in Equation 7-1. No default input values are presented because of the wide variability in subsurface soil conditions that affect constituent migration.

Dilution Factor Equation

Equation 7-2. $$DilutionFactor = 1 + \frac{Kid}{IL}$$

Where: $$d = (0.0112L^2)^{0.5} + d_a(1 - \exp\{(-LI)/(Kid_a)\})$$

And where:

K	=	Aquifer hydraulic conductivity (m/yr)
i	=	Hydraulic gradient (m/m)
I	=	Infiltration rate m/yr (recharge rate m/yr)
d	=	Mixing zone depth (meters)
L	=	Source length parallel to ground water flow (meters)
d_a	=	Aquifer thickness (meters)

IDEM will use the following criteria to evaluate submittals that calculate a site-specific dilution factor using Equation 7-2:

- ***K,*** the hydraulic conductivity, should be determined from the best available information. Consideration should be based on the following: an average of at least three slug tests, a grain-size analysis, published sources, pump test data, and calculation of constituent movement.

- ***i***, the hydraulic gradient, should be determined from at least three ground water wells (or piezometers), considering seasonal or other fluctuations.

- d_a, the aquifer thickness, must be based on the best available information and should always be accompanied by a competent and reasonable search of regional water well logs and should include well depths and their relation to aquifer thickness.

- ***I***, the infiltration rate or recharge rate, should be based on the best available information and should reference values from the Natural Resource Conservation Maps of the Soil Conservation Service or other published sources.

- ***L***, the source length, should be characterized at the source area. L is the greatest source length parallel to the ground water flow.

7.1.3 Site-Specific Data for the Soil Saturation Limit Equation

The soil saturation limit equation (see Equation 7-3) is used to calculate the site-specific constituent soil saturation limit, which may be appropriate when the closure level is limited by the default soil saturation level. The soil saturation limit (C_{sat}) corresponds to the constituent concentration in soil at which the following limits have been reached: (1) the adsorptive limits of the soil particles, (2) the solubility limits of the soil pore water, and (3) saturation of soil pore air. At concentrations that exceed the soil saturation limit, soil COCs may be present in free phase. The following site-specific factors may be determined and used in Equation 7-3:

- Dry soil bulk density
- Fraction of organic carbon (specific to surface or subsurface soil, whichever is appropriate)
- Water-filled soil porosity
- Air-filled soil porosity
- Soil particle density

7.1.4 Site-Specific Data for the Soil Attenuation Capacity Equation

The soil attenuation capacity equation (see Equation 7-4) allows the calculation of a site-specific soil attenuation capacity, which is one of the constituent source limits that must be evaluated for each discrete soil sample. The default soil attenuation capacity concentration is 6,000 milligrams per kilogram (mg/kg) total organic constituent for surface soil and 2,000 mg/kg total organic constituent for subsurface soil. The only site-specific factor used to calculate the soil attenuation capacity is the fraction of organic carbon specific to surface or subsurface soil (whichever is appropriate). A nondefault concentration may be calculated using Equation 7-4.

Soil Saturation Limit Equation

Equation 7-3. $$C_{sat} = \frac{S}{\rho_b}\left(K_d \rho_b + \theta_w + H'\theta_a\right)$$

Where:

C_{sat}	=	Soil saturation limit (constituent specific in milligrams per kilogram)
ρ_b	=	Dry soil bulk density (default at 1.5 kilogram per liter)
K_d	=	Soil-water partition coefficient in liters per kilogram where:
		K_d = K_{oc} x f_{oc}
		K_{oc} = Soil organic carbon partition coefficient (constituent specific in liters per kilogram)
		f_{oc} = Fraction organic carbon (default at 0.006 g/g)
θ_w	=	Water-filled soil porosity (default at 0.15 L water/L soil)
θ_a	=	Air-filled soil porosity (default at n - θ_w L air/ L soil)
n	=	Total soil porosity (1- ρ_b / ρ_s (L pore/L soil))
ρ_s	=	Soil particle density (default at 2.65 kg/L)
S	=	Solubility in water (constituent specific in milligrams per liter of water)
H'	=	Henry's Law Constant (dimensionless)

Soil Attenuation Capacity

Equation 7-4. *Site-Specific Soil Attenuation Capacity* = f_{oc} x 10^6

Where:

f_{oc} = Fraction of organic carbon in grams per gram

For example, 0.007 g/g fraction organic carbon x 10^6 = 7,000 mg/kg total soil attenuation capacity.

7.1.5 Site-Specific Data for Determining the Fraction of Organic Carbon

To determine the fraction of organic carbon (f_{oc}), soil samples must be collected from areas not affected by soil contamination to minimize interference from carbon-based constituents. Visual evidence, in conjunction with field screening and laboratory analyses, should be employed to locate areas not affected by constituents. Composite soil samples from at least two borings should be collected and analyzed separately to determine f_{oc}. The soil collected from these borings must be of similar nature and composition as the soil affected by the contamination.

If more than one soil type is present at depths corresponding to the vertical extent of soil contamination, separate composite samples that are representative of the soil variation should be collected A weighted average f_{oc} representing the affected area should then be calculated. If the vertical extent of soil contamination is significant, more samples should be included in the composites of each soil type; collecting additional samples will more accurately assess vertical soil variation. Weighted averages for f_{oc} can be calculated using Equation 7-5.

Weighted Averages for f_{oc}

Equation 7-5.

$$\bar{c} = \frac{\sum_{i=1}^{n} l_i c_i}{\sum_{i=1}^{n} l_i}$$

Where:

$\bar{c}$	=	Weighted average soil concentration
c_i	=	Representative soil concentration in an interval
l_i	=	Soil interval length
n	=	Interval number

No single method is recommended for analyzing f_{oc} ; however, the method should have a detection limit of 0.1 percent or less organic carbon; the soil-to-ground water partitioning model is not valid for soil that contains less than 0.1 percent organic matter. Some typical references for analytical methods for f_{oc} are presented below; some may apply to specific site conditions, such as glacial sediments:

- Allen-King, R. M., and others. 1997. "Organic Carbon Dominated Trichlorethene Sorption in a Clay-Rich Glacial Deposit." *Groundwater Journal*. Volume 35. Number 1. Pages 124 to 130.

- American Society for Testing and Materials. (1995). "Document D2974, Method C."

- Nelson, D. W., and Sommers, L. E. 1982. "Total Carbon, Organic Carbon, and Organic Matter." In *Methods of Soil Analysis. Part 2, Chemical and Microbiological Properties*. Second Edition. A.L. Page, editor. American Society of

Agronomy. Madison, Wisconsin. Volume 9. Number 2. Pages 539 to 579.

7.1.6 Site-Specific Data for Dry Soil Bulk Density and Soil Porosity

Discrete samples that are representative of the contaminated soil type must be collected to determine the dry soil bulk density and soil porosity. Because bulk density and porosity typically are not affected by most common constituents, sampling for these parameters may be possible in contaminated areas, depending on the constituent type and concentration. More than one sample per boring may be needed to completely characterize the soil. If more than one sample per boring is used, a weighted average should be calculated. If more than one soil type is present at depths corresponding to the vertical extent of contamination, the methodology outlined above for f_{oc} sample averaging should be followed. The following are commonly used method references for dry soil bulk density and porosity:

- American Society for Testing and Materials. 1996. "ASTM D2937." *Annual Book of ASTM Standards. Volume 4.08. Soil and Rock Building Stones*. Philadelphia, Pennsylvania.

- Klute, A. (editor). 1986. *Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods.* Second Edition. American Society of Agronomy. Madison, Wisconsin.

7.2 Plume Stability and Fate & Transport Modeling

Plume stability is an objective of ground water closure and may be demonstrated using default or nondefault methods. The default procedure for plume stability demonstrations (Appendix 3) provides a detailed mechanism to determine whether ground water degradation is occurring with respect to certain constituents. If the default stability monitoring process determines that a plume is expanding, the following options are available:

1. Use a remediation method such as sparging, pump and treat, or monitored natural attenuation, as appropriate

2. Evaluate plume stability using a nondefault process

The subsections below discuss the nondefault process for demonstrating plume stability and the use of fate and transport modeling.

7.2.1 Nondefault Plume Stability

In some situations, it may be appropriate to make minor adjustments to the default stability monitoring procedure. Such adjustments are considered nondefault variations to the default procedure and will require IDEM program approval. Examples of nondefault variations include the following:

- Using existing historical ground water data that is incomplete (for example, if most ground water data is appropriate for the default procedure but some quarterly data may be missing)

- Proposing additional monitoring, reassessment, and evaluation using the default Mann-Kendall Test if a low percentage of sample results exceed closure levels

Other nondefault plume stability demonstrations may involve more rigorous methods; these would primarily involve alternate statistical evaluations that replace the Mann-Kendall evaluation as well as fate and transport modeling.

7.2.2 Fate and Transport Modeling

Fate and transport models may be useful in modeling potential constituent transport from one medium to another. Such models may also be useful in estimating constituent concentrations (either temporally or geographically) when sampling data are not available. IDEM anticipates that fate and transport modeling will be proposed for the following purposes:

- To evaluate potential exposure pathways (for example, to estimate possible ground water concentrations based on the soil-to-ground water pathway or to estimate possible air concentrations based on the soil-to-air pathway)

- To estimate possible constituent concentrations at different downgradient points (for example, based on ground water data collected at upgradient locations)

- To estimate the timeframe required for the following:

  - Meet applicable risk-based objectives
  - Complete the remedial action
  - Achieve plume stability
  - Achieve closure

- To demonstrate the effectiveness of a given remedial action plan or closure plan to meet applicable risk-based objectives

- To support other risk-based determinations, as appropriate

Fate and transport modeling involves two key determinations: (1) selecting a model appropriate for the situation and (2) selecting values for the model input parameters. In a nondefault submittal, the risk assessor may propose fate and transport models and appropriate input values specific to the source area and the model. IDEM will evaluate these submittals based on relevant EPA guidance and the following criteria and ASTM publications:

- Appropriateness for the site-specific conditions given the limitations inherent in the model

- Availability of sufficient data

- Adequacy of documentation

- Proper calibration, including sensitivity or error analyses

- Correct use of support assumptions about future conditions

- The following ASTM publications:

  - Standard Guide for Comparing Ground-Water Flow Model Simulations to Site-Specific Information (ASTM D5490-93e1)

  - Standard Guide for Documenting a Ground-Water Flow Model Application (ASTM D5718-95e1)

  - Standard Guide for Application of a Ground-Water Flow Model to a Site-Specific Problem (ASTM D5447-93)

– Standard Guide for Conducting a Sensitivity Analysis for a Ground-Water Flow Model Application (ASTM D5611-94e1)

– Standard Guide for Calibrating a Ground-Water Flow Model Application (ASTM D5981-96e1)

– Standard Guide for Defining Boundary Conditions in Ground-Water Flow Modeling (ASTM D5609-94e1)

– Standard Guide for Describing the Functionality of a Ground-Water Modeling Code (ASTM D6033-96)

– Standard Guide for Defining Initial Conditions in Ground-Water Flow Modeling (ASTM D5610-94e1)

– Standard Guide for Documenting a Ground-Water Modeling Code (ASTM D6171-97)

– ASTM Standards on Determining Subsurface Hydraulic Properties and Ground Water Modeling, 2nd Edition, International Standard Book Number (ISBN) 0-8031-27170

For all nondefault plume stability demonstrations, the appropriateness of other methods and the ensuing monitoring period will depend on site conditions, complexity, and the limitations of the approach.

7.3 Modifying Exposure Assumptions

A nondefault evaluation offers enough flexibility to tailor exposure assumptions to site-specific conditions or to modify exposure assumptions based on current peer-reviewed research. In a nondefault evaluation, industrial exposure equations and assumptions may be modified based on site-specific factors. However, IDEM considers potential long-term residential land use activities to be similar everywhere. Nevertheless, IDEM will consider changes to the residential exposure assumptions based on new and compelling information. Such changes will be "permanent" changes to the default approach and will be applied statewide and not on a site-specific basis. IDEM will evaluate submittals that propose modified exposure assumptions based on the following criteria:

- EPA acceptance

- Consistency with evaluation of Reasonable Maximum Exposure
- Reliance on institutional controls for limiting exposure
- Relative uncertainty
- Applicability and relevance

7.4 Institutional Controls

Two default institutional controls are available in RISC: (1) an Environmental Notice and (2) the demonstration of an appropriate ground water ordinance. However, IDEM recognizes that other mechanisms may reasonably accomplish the desired exposure control in a manner consistent with the default mechanisms. Nondefault institutional controls proposed as part of site closure will be evaluated against the following criteria:

- The control provides ~~legal~~ constructive (7-24-01) notice to current and future owners of the affected property of the nature and extent of the restrictions.

- The control is permanent in nature.

- The control is legally valid.

Nondefault institutional controls may be approved if they satisfy these criteria.

7.5 Considering Other Pathways, Exposures, and Media Not Included in the Default

The sections presented above describe modifications that may be made to default equations and models to reflect site-specific conditions. Other deviations from the default approach may be necessary because it may not address all appropriate pathways, exposures, and media. For example, risk-based closure criteria were calculated for soil and water media because constituent behavior in these media is generally well understood and easily measured. However, air pathways (including ambient outdoor air, odors, vapor intrusion through basements, and indoor air from sources other than basements) were not evaluated in the default approach.

RISC does not specifically offer guidance on how to evaluate these pathways, exposures, and media; nevertheless, they should be evaluated as appropriate because they may pose a significant risk at contaminated sites. This section is intended to identify those concerns not otherwise addressed in the default approach. Default exposure pathways and exposure routes are presented in Table 7-2.

Table 7-2. Default Exposure Pathways and Routes

Exposure Pathway	Residential Land Use	Commercial or Industrial Land Use	Construction Worker Exposure
Direct Soil Contact	▪ Skin contact ▪ Ingestion of soil ▪ Inhalation of soil vapors and particulates		
Soil Leaching to Ground Water	▪ Ingestion of ground water contaminated by soil leachate		▪ Not evaluated
Ground Water	▪ Ingestion of ground water ▪ Inhalation of vapors released from ground water	▪ Ingestion of ground water	▪ Not evaluated

A nondefault evaluation must be used to assess current and future exposure pathways that are not addressed in the default approach. The default approach makes certain assumptions about land use, potential pathways, and routes of exposure. When site conditions fall outside the scope of the default approach, then a nondefault risk analysis must be performed. It is erroneous to assume that only default media and pathways require evaluation; nondefault media or pathways should also be evaluated if appropriate. Generally, determining exposure scenarios requires that four types of media be considered:

- Air
- Soil
- Ground water
- Surface water (and sediments)

Air exposure can occur through ambient outdoor air or indoor air. Soil exposure can occur by ingestion, dermal absorption, and inhalation of volatiles and particulate matter (integrated into one closure level — direct contact). Constituent migration from soil to ground water may result in ground water exposure. Ground water and surface water exposures may result from drinking water, bathing, cooking, or industrial process applications such as cooling. Surface water evaluations should also consider sediments because they may be contaminated with constituents that tend to partition out of surface water. In particular, sediments should be evaluated for bioaccumulative ~~COCss~~ COCs (7-24-2001) because they tend to partition into sediment, where they then may enter the food web.

Biota exposures (including potential plant and animal COC uptake, with subsequent human consumption) should also be considered for each medium, if appropriate. Biota exposure pathways merit consideration when consumable plants and animals in the area may be affected. Other factors to be considered include the following:

- Constituent deposition from air to plants that are ultimately consumed
- Constituent deposition from air to surface water and soil
- Uptake through irrigation with contaminated water
- Uptake through livestock watering with contaminated water
- Consumption of forage grown on contaminated soil
- Aquatic species uptake in contaminated water and sediments

For example, if an area has been affected by constituents and it is used for grazing dairy cattle, it would be appropriate to evaluate plant uptake of the constituent for bioaccumulation in the dairy herds. As another example, if surface water has been contaminated, subsistence fishing should be evaluated to determine if this is a viable pathway for exposure. Additional information on indirect exposures is provided in *Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities* (1994).

Evaluating the applicability of these pathways and exposure routes is important in the nondefault approach. Potential exposure media and some of the associated nondefault pathways are summarized in Table 7-2. The vapor intrusion pathway is particularly important when volatiles are present in the vicinity of basements. Guidance on the evaluation of this and other air pathways is available in the *Soil Screening Guidance: Technical Background Document* (EPA 1996). Consultation with IDEM is recommended if any nondefault pathway must be evaluated.

Table 7-3 Nondefault Exposure Media and Associated Pathways

Exposure Media	Examples of Associated Pathways
Soil	▪ Runoff to surface water ▪ Vapor intrusion ▪ Biota – Commercial produce consumption – Plant uptake associated with meat, dairy, and game
Ground water	▪ Industrial process water ▪ Biota uptake in irrigated produce
Air	▪ Ambient – Particulate – Vapors ▪ Indoor air ▪ Particulate deposition on soil ▪ Biota uptake from air deposition on plants and soil ▪ Uptake by aquatic plants and animals from air deposition on surface waters
Surface water	▪ Recreational ▪ Drinking water ▪ Sediments ▪ Biota – Benthic uptake from sediments – Fish consumption – Uptake by irrigated produce

Determining which pathways or media can be eliminated from further consideration is largely a matter of investigating the potential for exposure. In a nondefault, site-specific approach, exposure pathways may be eliminated from further consideration with adequate justification. For instance, with appropriate institutional controls in place, the following pathways might be eliminated:

- Direct contact pathways for surface and subsurface soil if an asphalt surface or other barrier approved by IDEM effectively prevents direct contact with contaminated media

- Recreational exposure pathways in surface water if swimming and related exposures are prohibited or are demonstrated to be unrealistic

- Ingestion pathways for ground water

The evaluation criteria for submittals that rely on pathway elimination will include the following:

- Evidence for current exposure

- Potential for future exposure
- Effectiveness of institutional controls
- Overall protectiveness of the remedy

7.5.1 Source Areas Larger than ½ Acre

In screening, source areas larger than ½ acre may be partitioned into ½ acre increments to evaluate surface soil using the default Max and Chen procedures (see Chapter 3). After surface soil samples are collected in each ½ acre partitioned area (according to requirements of the Max or Chen tests), analytical results can be evaluated for each partitioned acre.

For characterization and closure sampling, the source area could similarly be partitioned into ½ acre increments. Each ½ acre increment could then be sampled in 10 locations to establish a concentration gradient; in this manner the perimeter of the entire source area could be established through sampling. As with screening, this sampling approach for characterization and closure could result in the collection and analysis of a large number of samples. More cost-effective strategies may be possible. Section 7.9.3 provides sampling strategies that may reduce the number of samples needed for larger source areas.

The default soil-to-ground water partitioning model used to evaluate constituent migration to ground water incorporates a dilution attenuation factor of 20 for source areas up to ½ acre. Any source area larger than ½ acre must be evaluated using a smaller DAF if the soil-to-ground water partitioning model is used. A dilution attenuation factor of 10 may be appropriate for source areas up to 30 acres in size, or a more appropriate model may be proposed to evaluate this pathway. IDEM recommends that regulatory support be sought from the appropriate remedial program for source areas larger than 30 acres.

7.5.2 Karst and Fractured Flow Geology

Karst terrain and fractured flow geology will require a nondefault approach to closure for (1) the soil constituent migration to ground water pathway and (2) the ground water ingestion pathway. Unconsolidated materials overlying fractured flow areas may be evaluated using default strategies for direct contact pathways. However, the soil-to-ground water partitioning model is not valid if applied to consolidated, heterogeneous, and nonisotropic materials. In such cases, the evaluation of the migration to ground water pathway will require a nondefault approach that should be discussed with IDEM.

For the ground water ingestion pathway, the following activities are always site specific in karst terrains:

- Screening
- Delineating the nature and extent of contamination
- Closure sampling
- Plume stability demonstrations

The development of any ground water sampling strategy in fractured flow geology will require close coordination with IDEM to most efficiently address the uncertainty associated with these endeavors.

7.5.3 Impacts on Ecologically Susceptible Areas

For ecologically susceptible areas, a nondefault assessment is required if constituents are identified within these areas, or if contaminated areas are connected to such areas by an exposure pathway (for example, surface runoff connecting a source area to a wetland, stream, or lake). The following guidance may be helpful in conducting ecological risk assessments:

- Ecological Risk Assessment and Risk Management Principles for Superfund Sites (1999) - Available online at http://www.epa.gov/superfund/programs/risk/tooleco.htm

- Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, Interim Final (1997) - Available online at http://www.epa.gov/superfund/programs/risk/ecorisk.htm

- Guidelines for Ecological Risk Assessment (1998) - Available online athttp://www.epa.gov/ncea/ecorsk.htm

- The Wildlife Exposure Factors Handbook - Available online at http://www.epa.gov/ncea/wefh.htm

- U.S. Fish and Wildlife Service provides information related to ecological risk assessment. The information is available at http://contaminants.fws.gov

7.5.4 Exposures of Acute or Subchronic Duration

The default approach does not address exposures of acute or subchronic duration. If these exposures are appropriate for evaluation (for example, air pathways), a meeting with IDEM is suggested before any proposal is submitted.

7.6 Sampling Soil and Ground Water

Chapters 3, 4, and 6 describe default sampling methods for screening, characterization of the nature and extent of contamination, and closure, respectively. Other methods or variations on the default methods may also be reasonable alternatives to the default guidance provided in those chapters. The purpose of this section is to offer guidance on nondefault approaches for sampling ground water and soil. Possible nondefault approaches and associated criteria are presented in Table 7-4.

See Section 7.9.4 for additional information on random sampling within a grid system.

7.7 Carcinogen Target Risk Level

IDEM has established a default target risk of 10^{-5} as protective of human health, when used in conjunction with the default equations, toxicity criteria, and measurement of potential exposure concentrations (PEC). In establishing the default, IDEM has defined a risk assessment that is generally applicable anywhere in the state. The level of uncertainty associated with PEC determinations and assumptions made in the default approach have been determined to be acceptable.

In order to deviate from the default target risk level associated with this predetermined level of uncertainty, a new site-specific risk assessment must be performed. In a nondefault risk assessment, IDEM will evaluate target risk proposals within the 10^{-4} to 10^{-6} risk range in a manner consistent with the National Contingency Plan (NCP) and EPA Office of Solid Waste and Emergency Response Directive 9355.0-30 “Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions.”

Table 7-4. Nondefault Sampling Criteria

Procedure	Nondefault Criteria
Screening surface and subsurface soil	1. Sample in a manner that provides sufficient information to determine if additional investigation is warranted for the area in question. 2. Evaluation criteria for the sampling data should be consistent with the sampling method. Judgmental samples are generally compared individually to the closure level. A statistical evaluation is performed on randomly selected samples.
Ground water screening	1. Sample in a manner that provides sufficient information to determine if additional investigation is warranted for the area in question.
Characterize the nature and extent of soil contamination	1. Collect an adequate number of samples to reasonably characterize constituent concentrations across the source area. 2. Evaluation criteria for the sampling data should be consistent with the sampling method. Judgmental samples are generally compared individually to the closure level. A statistical evaluation is performed on randomly selected samples. 3. For large, complex sites with multiple source areas, characterize each source area to where the extent of contamination meets the industrial closure level. Next, collect samples at the property line to determine if off-site areas have been contaminated at concentrations that exceed residential concentrations. Sampling at the property line may be conducted in lieu of delineating each source area laterally to where the extent of contamination meets residential concentrations.
Sampling soil for closure	1. Collect an adequate number of samples to reasonably characterize constituent concentrations across the source area. 2. Evaluation criteria for the sampling data should be consistent with the sampling method. Generally, judgmental samples are not appropriate for closure sampling if remediation has been performed.
Characterize the nature and extent of ground water contamination	1. Collect an adequate number of samples to reasonably characterize constituent concentrations across the source area.
Sampling ground water for closure	2. Collect an adequate number of samples to reasonably demonstrate that closure levels have been achieved. 3. To determine plume stability, collect samples in a manner consistent with the requirements of the appropriate ground water fate and transport model.

Any changes to the default exposure assumptions, pathways, PEC procedures, or other factors used in the risk assessment may introduce greater uncertainty. This uncertainty should be evaluated either quantitatively, qualitatively, or both, and a decision should be made regarding the acceptable level of uncertainty to support the selection of a target risk.

- If the default pathway and additivity equation (see Appendix 1, Table C); default toxicity (see Table F); and default exposure criteria (see Table D); and default chemical constants (see Table B) are used, the default target risk level (10^{-5}) may be used without additional justification.

- If changes are proposed to any default pathway or additivity equation inputs, toxicity, or exposure criteria, the evaluation should select a target risk level within 10^{-4} to 10^{-6} based on the level of uncertainty introduced by the proposed change.

- When evaluating the appropriateness of a proposed target risk IDEM will consider uncertainties in a nondefault risk assessment for either multiple or single constituents. The more uncertainty that can be eliminated, the greater the consideration of a higher target risk level (10^{-4}). The more uncertainty that is added, the greater the consideration of a lower target risk level (10^{-6}).

IDEM believes this approach best incorporates the decision-making process associated with Superfund and IDEM's broad application of risk-based decision making across all cleanup programs.

7.8 Noncarcinogen Additivity Approach

Nondefault approaches can be taken by demonstrating that constituent effects are limited to a single organ, or that the toxic effects occur by separate, nonadditive mechanisms. Nondefault approaches will be assessed on a case-by-case basis and evaluated against a hazard index of 1.0.

7.9 Nondefault Characterization and Closure Sampling

Hazardous waste sites may cover several acres of land; however, ~~constituent~~ contaminant (7-24-2001)source areas may be smaller than ½ acre and could be managed using the default procedures outlined in Chapters 1 through 6. The following procedures may be used for any source area, including contaminant source areas greater than ½ acre.

While the procedures outlined here have been found to be suitable at many sites, not all sites will be able to use every method. It is likely that greater interaction with IDEM technical staff will be required when selecting and using nondefault methods.

In general, the site is first divided into distinct source areas (horizontal strata), then a sampling plan is developed for each stratum, samples are collected, and analytical data are evaluated statistically. A broad look at hazardous waste closure decisions identifies the following activities as necessary steps in this process:

- **Define the sample area**. The waste site should be divided into sample areas (horizontal strata). Each sample area will be evaluated separately for attainment of closure levels and will require a separate statistical sample. It is important to ensure that sample areas are clearly defined during the data quality objectives (DQO) process (See Appendix 6).

- **Specify the constituents for which to test**. Constituents to be tested for in each soil unit should be listed (See Chapter 5).

- **Specify the sample handling and collection procedures**. An important task for any decision procedure is to define carefully how each parameter will be sampled and analyzed.

- **Establish the closure level.** Closure levels are determined by site-specific risk assessments, by guidance, or by rule (see Chapter 6.

- **Specify the parameter (statistic) to be compared to the cleanup standard**. For RISC we use the upper confidence limit of the mean for each stratum.

- **Specify the probability of mistakenly declaring the sample area clean**. Select and specify the false positive rate (see Glossary) for testing the site. It is recommended that all constituents in the sample area use the same rate. This rate is the maximum probability that the sample area will mistakenly be declared clean when it is actually dirty.

The following sections outline the process for determining whether the desired environmental concentrations have been attained and the site is eligible for closure:

1. Presampling activities
2. Horizontal stratification

3. Sample size determination
4. Selection of sample locations
5. Sample collection and analysis
6. Data quality assessment

7.9.1 Presampling Activities

As in the RISC default approach, the nondefault process is designed to achieve a high level of confidence in source area characterization by completing thorough presampling activities. Presampling activities include a review of site information, selecting an approach to sampling, determining the boundaries of the waste site, and obtaining or preparing a detailed map of the waste site (see Chapter 2). It may be advantageous to do some preliminary sampling during this step. The data obtained can be used to help develop an accurate approximation of the required sample size.

7.9.2 Horizontal Stratification

Unless constituents and concentrations are homogeneous throughout the entire site, the site must be stratified into source areas with similar characteristics.

Three key terms describe areas within the waste site:

- Source area
- Horizontal strata
- Sample location

For the purposes of this guidance manual, source areas and horizontal stratum generally identify surface areas designated for sampling. Subsurface samples are taken from vertical strata below the horizontal stratum. Because ground water is mobile, ground water samples may be required outside the area of soil contamination. A sample location is the point within an individual stratum at which one takes a sample. See Chapter 2 for more information on classifying areas of a site and developing a conceptual site model (CSM).

Proper stratification of a hazardous waste site ensures that samples are grouped to meet the project objectives of site characterization and closure in an effective yet efficient manner. The precision of statistical estimates is likely to be improved by dividing a large site into homogeneous strata. In this way, the variability due to soil, location, characteristics of the terrain can be controlled, thereby improving the precision of contamination level estimates.

The following can be used to define horizontal and vertical strata in an area:

- Sampling depth
- Constituent concentration
- Physiography and topography
- The presence of interferants that affect laboratory analytical techniques
- The history and sources of contamination at the site
- Previous cleanup attempts
- Weathering and run-off processes

Two concepts are central to the process of separating an area into strata:

- The strata must not overlap; no area within one stratum can be within another stratum (See Figure 7-1).
- The sum of the sizes of the strata must equal the total area to be evaluated.

Site characterization decisions should be made independently for each source area. It is important to ensure that source areas are clearly defined and agreed to by all. It is generally useful to define multiple source areas (horizontal strata) within a waste site or source area. These areas should be defined so that they are as homogeneous as possible with respect to prior waste management activities. For example, if a PCB transformer disposal area and a lead battery recycling area are located on the same site, they should generally not be included in the same source area unless contamination from the two sources overlaps. In that case, there would be three separate source areas for sampling, the PCB-contaminated area, the lead contaminated area, and the area contaminated by both.

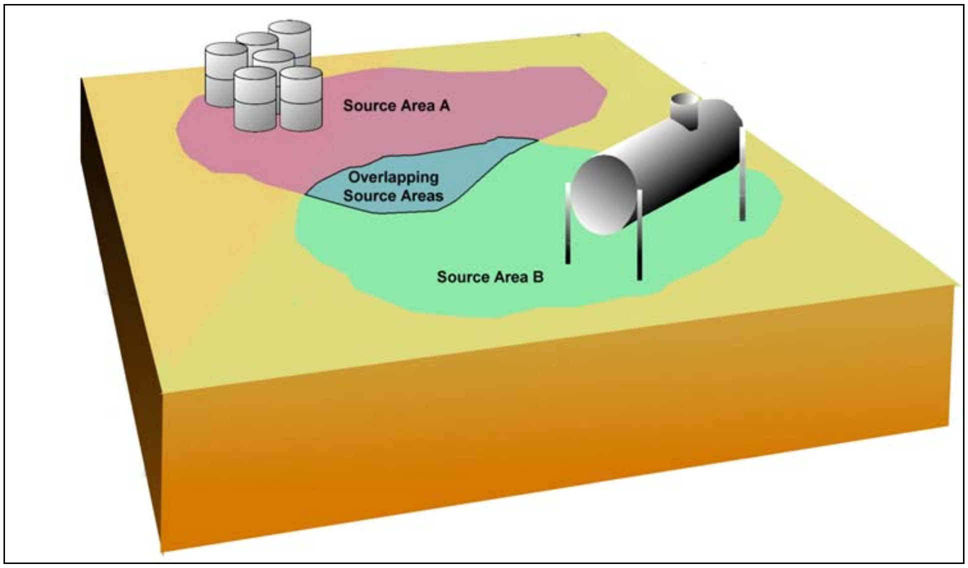


Figure 7-1 Overlapping Source Areas

Additionally, a site may be comprised of areas that require different sampling or treatment technologies. For example, disturbed versus natural soils, wetlands versus firm terrain, or sandy versus clay soils may suggest establishment of different sampling areas for stratification. Sample area definitions also require that the depth or depth intervals of interest be specified. Section 3.4.3 provides information on subsurface sampling.

7.9.3 Sample Size Determination

This section discusses types of sample size for closure and other useful calculations related to sample size.

7.9.3.1 Types of Samples

Three types of samples are taken in the RISC process: screening samples, characterization samples, and closure samples. RISC allows screening and characterization samples to be used for a closure decision under the following circumstances:

1. The samples are taken using a method that will determine the worse case scenario at the site, and all sample concentrations are less than the established cleanup level

2. Samples were collected in such a way that the sample mean is representative of the entire source area, and the upper confidence limit of the mean is below the closure level.

Guidance on a worst-case characterization is given in Section 3.4.1. In all cases, for source areas ½ acre or greater, the sample number will not be less than 10 characterization samples plus 4 samples (one each from upgradient, downgradient, and the two side gradient directions) that have constituent concentrations less than closure levels.

7.9.3.2 Calculating the Sample Size for Closure

Determining the appropriate sample size to make a closure decision requires some knowledge of the concentrations of environmental constituents for the source areas at the site. This can be acquired through preliminary sampling. Often this data is not available. The following two options can be used when no data are available:

1. Use the calculation $\sqrt[3]{n}$ (n = number of grid points in the sample area) as an initial sample number estimate, or

2. Estimate the mean and standard deviation for use in **Equation 7-6** below

Careful consideration must be given to which values are used. If the calculated sample size is too small, you will be required to obtain additional samples, if it is too large, costs increase unnecessarily.

The following equation is used to determine the appropriate sample size. Initially an estimate is calculated using **Equation 7-6**. When samples have been collected and analyzed, the calculation is repeated using the mean and standard deviation from the full data set to determine whether the sample size is adequate to make a closure decision.

It is advisable to collect and properly store a few additional samples while at the site, paying careful attention to preservation and maximum holding times. Then, if a final sample size calculation shows a need to analyze additional samples, the additional stored samples are available for analysis and the need to remobilize for sampling is limited. *The minimum sample size (definitive samples sent to the laboratory) for source areas greater than or equal to ½ acre will never be less than 14.*

Sample Size Calculation

Equation 7-6.

$$n = \frac{(Z_\alpha + Z_\beta)^2}{(C_s - \mu_1)^2}\sigma^2 + \frac{Z_\alpha^2}{2}$$

Where:

n	=	The required number of samples
Z_α	=	The "Z" value for the selected alpha (α = Type I error, 1 - α = confidence limit)
Z_β	=	The "Z" value for the selected beta (β = Type II error, 1 - β = power)
C_s	=	The acceptable constituent level at the site
μ_1	=	The population mean at the site (often estimated by the sample mean)
σ^2	=	The population variance (often estimated by the sample variance)

Note: α and β are selected based on decision error limits

The sample number required varies depending on the standard deviation and the constituent level found at the site. As would be expected, if the site mean is close to the limit for a constituent, and the standard deviation is large, a large number of samples is required to provide confidence that the actual site mean (μ_1) is below the limit. EPA/600/R-96/084, Guidance For Data Quality Assessment.

7.9.3.3 Other Useful Calculations Related to Sample Size

The coefficient of variation and the calculation of the action level are two other useful calculations related to sample size. The closer the sample mean is to the closure level and the greater the variability in the sample data (large standard deviation), the greater the sample number required to confirm that the closure level is not exceeded. If the approximate mean and standard deviation are known it is possible to use a "rule of thumb" to evaluate whether a large sample number will be required. Population parameters are estimated using sample data.

Coefficient of Variation

Equation 7-7. $CV = \frac{\sigma}{\mu_1}$

Where:

σ	=	Population standard deviation
μ_1	=	Population mean

For the sample number to be reasonable, the coefficient of variation (CV) must satisfy one of the following conditions:

- CV ≤ 0.5 if the sample mean is ∃0.7 of the limit
- CV ≤1.0 if the sample mean is ∃0.45 but #0.7 of the limit
- CV ≤1.5 if the sample mean is #0.45 of the limit

As the mean increases the CV must get smaller or the sample size must increase.

Consider the following two examples:

Closure Level = 72 parts per million (ppm)
mean = 90% of the limit or closure level
$\mu_1 = 64.8$ ppm
$\sigma = 32.4$
mean = 80% of the limit
$\mu_1 = 57.6$ ppm
$\sigma = 28.8$

The 90% values σ and μ_1 given above (CV = 0.5) yield a sample number of about 127 as follows:

$$n = \frac{(1.645 + .842)^2}{(72 - 64.8)^2}(32.4)^2 + 1.35 \approx 127$$

If the site mean is 80% of the closure level with CV = 0.5, the sample number is about 25.

$$n = \frac{(1.645 + .842)^2}{(72 - 57.6)^2}(28.8)^2 + 1.35 \approx 26$$

When the site mean is close to the limit the required sample number is high, even with a fairly low coefficient of variation. A smaller standard deviation will result in a smaller number of required samples, but hazardous material cleanup sites often have significant variations in the constituent concentration of samples, and because of this a high standard deviation. This is especially true prior to any remedial efforts.

If preliminary sampling indicates that the site concentrations are near the limit it may, in the long run, be more cost effective to perform cleanup activities before attempting closure of the source area.

The calculation of the action level can be quite useful. In the example above (in which the site mean is at 90 percent of the acceptable constituent level and only 25 samples are taken) site concentrations exceed the action level. In the other example (in which the site mean is at 80 percent of the acceptable level) site concentrations are less than the action level, and 25 samples were adequate to establish that the site mean probably does not exceed the regulatory limit. These calculations are tools to help determine the approximate cleanup level and sample number required to close a site.

ASTM (D5792) Calculation for the Action Level

Equation 7-7. $$AL = RT - Z_{0.05}\frac{S_w}{\sqrt{n}}$$

Where:

AL	=	The action level
RT	=	The regulatory threshold
$Z_{0.05}$	=	The number from the Z table corresponding to the 95% confidence level
S_w	=	The sample standard deviation
n	=	The sample number

Using the examples above:

$$AL = 72 - 1.645\frac{28.8}{\sqrt{25}} = 62.5\,ppm$$ = action level for Φ = 28.8 and 25 samples

$$AL = 72 - 1.645\frac{32.4}{\sqrt{25}} = 61.3\,ppm$$ = action level for Φ = 32.4 and 25 samples

7.9.4 Selecting Appropriate Sample Locations

At a minimum, the following activities should be included in the sampling design:

- Review existing historical site information
- Perform a site reconnaissance
- Evaluate potential migration pathways and receptors
- Determine sampling objectives
- Establish DQOs
- Collect field screening data
- Select parameters for which to analyze
- Select an appropriate sampling approach
- Determine sampling locations

Randomization is necessary to make probability or confidence statements about the sampling results. Sample selection using the judgment of the sampler has no randomization. Results from such samples cannot be generalized to the whole sample area, and no probability statements can be made when judgmental sampling is used.

However, judgmental sampling may be justified, for example, during the preliminary assessment and site investigation stages, if the sampler has substantial knowledge of the sources and history of site contamination. However, judgmental samples should not be used to determine if the cleanup standard has been met (except as outlined above and in Chapters 3 and 6).

Random selection of sample points requires that each sample point be selected independent of the location of all other sample points. With random sampling, no pattern is expected in the distribution of the points. However, it is possible (purely by chance) that all of the sample points will be clustered in one or two quadrants of the site. This possibility is extremely small for larger sample sizes.

An alternative to random sampling is systematic sampling, which distributes the sample more uniformly over the site. A random starting point is selected, and samples are collected in a pattern covering the entire source area. Because the sample points follow a simple pattern and are separated by a fixed distance, locating the sample points in the field may be easier using a systematic sample than using a random sample. In many circumstances, estimates from systematic sampling may be preferred. More discussion of systematic versus random sampling can be found in *Finney (1948), Legg, et al. (1985), Cochran (1977), Osborne (1942), Palley and Horwitz (1961), Peshkova (1970), and Wolter (1984)*. Complete references for these sources are provided at the end of this chapter.

The procedures outlined below should ensure the following:

- The method of establishing soil sample locations in the field is consistent with the planned sample design.
- Each sample location is selected in a nonjudgmental and unbiased way.
- Complete documentation of all sampling steps is maintained.

The procedures assume that the sampling plan has been selected, the boundaries of the source areas and any strata have been defined, a detailed map of the waste site is available, and the required sample size is known.

Soil screening borings and sampling areas should be located in a manner that can determine with a high level of confidence if any previously specified constituents are present. Random sampling may be performed in a grid system. Judgmental sampling using default procedures should be performed at areas of suspected contamination, such as cracked areas of a containment structure, areas of known spills, or suspected downslope, downwind, or runoff areas of a containment structure. Other directed or systematic sampling methods (such as sampling at uniform intervals) may be used if warranted on a site-specific basis. These methods may include a (1) circular pattern of sampling around a central point or (2) linear sampling along a drainage way, boundary, or perimeter of a container storage area.

7.9.4.1 Selecting the Sample Coordinates for a Simple Random Sample

A random sample of soil units within the sample area or stratum will be selected by generating a series of random (X,Y) coordinates (pairs), finding the location in the field associated with these (X,Y) coordinates, and following proper field procedures for collecting soil samples. If the waste site contains multiple horizontal strata, the procedure described here is used to generate random pairs of coordinates for each stratum. The number of soil samples to be collected must be specified for each stratum.

Establish a square or triangular grid pattern inside of a rectangle which covers the entire sampling area, then generate random coordinates (Xi,Yi) which will be the locations of the sample points.

For a systematic sample, the size of the sample area must be determined in order to calculate the distance between the sampling locations in the systematic grid. The area can be measured on a map using a planimeter. The units of the area measurement (such as square feet, hectares, square meters) should be recorded. In areas suspected of being contaminated the grid size is seldom greater than ten feet.

For random sampling, a grid can be set up using professional judgement. For each stratum determine the shortest interval between two points which would provide reasonably independent samples. Generally the distance is shorter in high concentration areas and longer in low concentration areas. Establish a grid with this distance as the grid size (for example, 10 feet between grid lines).

The sample coordinates (Xi,Yi) can be generated using a random number generator. If random numbers are generated which fall outside the range of coordinates within the stratum they are ignored.

7.9.4.2 Field Procedures for Determining the Exact Sampling Location

The grid points specified for the coordinate system provide the starting point for locating the sample points in the field. The location of a sample point in the field will be approximate because the sampling coordinates were rounded to distances that are easy to measure, the measurement has some inaccuracies, and there is judgment on the part of the field staff in locating the sample point.

A procedure to locate the exact sample collection point is recommended to avoid subjective factors that may affect the results. Without this precaution, subtle factors such as the difficulty in collecting a sample, the presence of vegetation, or the color of the soil may affect where the sample is taken, and thus bias the results.

7.9.4.3 Sampling Across Depth

Methods for deciding how and where to subsample a soil core are important to understand and include in a sampling plan. These methods should be executed consistently throughout the site. The field methods used will depend on many things including the soil sampling device, the quantity of material needed for analysis, the constituents that are present, and the consistency of the solid or soils media that is being sampled. The details of how these considerations influence field procedures are not the subject of this discussion, but they are important. More detail can be obtained in Chapter 6 and the Soil Sampling Quality Assurance User's Guide (EPA 1984).

7.9.4.4 An Example of the Simple Grid Sampling Procedure

The following example illustrates a very simple grid sampling procedure:

1. Establish a grid that slightly overlaps the area to be sampled. The grid should not be limited to the boundaries of the area unless sampling would be obstructed by a building or other barriers. The grid interval may vary from site to site, but it will seldom be greater than 10 feet.

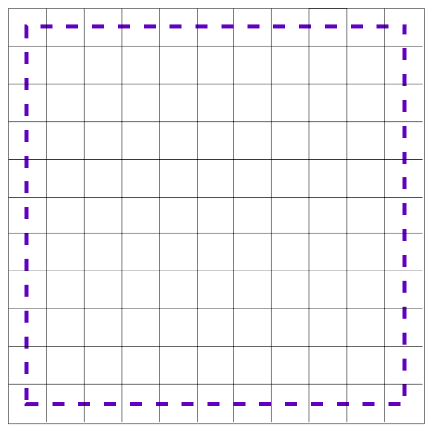
2. The number of sample borings required to adequately screen an area is determined by the sample size calculation (see Section 7.9.3.2). The minimum number of borings is three. The grid interval or number of sample borings may be modified if IDEM agrees that site-specific conditions warrant such changes.

3. Number the grid intersections and use a random number generator to determine which grid points will be sampled (see Section 7.9.4.1). Random numbers that indicate grid points outside the source area should be regenerated.

~~4.~~ Any proposal that includes this approach should include detailed drawings of the grid depicting sample locations.

EXAMPLE

The storage pad dimensions are 100 feet by 100 feet.
The grid interval is 10 feet.
The grid overlaps the pad by 5 feet on each side.
There are 144 grid intersections.
The number of borings should be equal to the cube root of 144.
($144^{1/3}$ = 5.2 or 6)



7.9.4.5 Ranked Set Sampling

Typically the most expensive part of the site evaluation process is laboratory analysis, while identification of potential sample units is a comparatively simple matter. We can therefore achieve great observational economy if we are able to identify a large number of sample units to represent the population of interest, yet only have to quantify a carefully selected subsample.

Ranked Set Sampling (RSS) is a method which can produce a better estimation of the site mean with the same number of observations, or an equal estimation of site mean with fewer observations. This can result in a significant reduction of costs. Since there will be fewer observations than with a simple random sample, correct ranking of potential laboratory samples is very important.

When is RSS Allowed?

The use of ranked set sampling is allowed under the following conditions:

The constituent distribution within a stratum is (1) continuous in nature; (2) not from isolated sources, such as buried drums or transformers; and (3) distributed throughout the stratum.

An acceptable ranking mechanism is available that (1) measures the appropriate constituents or (2) is an acceptable surrogate for the constituents and is accurate enough to correctly rank sets of samples.

Any statistical analysis performed is appropriate for the data distribution.

A Simple Example

As a simple introduction to the concept of RSS, consider the following example.

We wish to estimate the mean height of students at a university from a random sample of three students. Furthermore, to acknowledge the inherent uncertainty, we need to present this estimate as a confidence interval within which we expect the true population mean to lie with desired confidence.

The simplest way to obtain our sample is to randomly select three students from the university's population, then measure their heights. While the arithmetic average of the three heights is an unbiased point estimate of the population mean, the associated confidence interval can be very large, reflecting the high degree of uncertainty with estimating a large population mean with only three measurements. This is because we have no control over which individuals of the population enter the sample. For example, we may happen to grab two very short people and one very tall; or we may grab three very tall people. The only way to overcome such a problem with a simple random sample (SRS) is to increase the sample size.

On the other hand, we may obtain a ranked set sample. To do this, we may randomly invite three students to breakfast and visually rank them with respect to height. We then select the student we believe is shortest and actually measure his or her height. Repeating this process with lunch, we select the middle ranked person, and at dinner select the tallest ranked person. The resulting measurements of student heights constitute a ranked set sample. As with the SRS measurements, the arithmetic average of the RSS measurements provides an unbiased point estimate of the population mean; however, the associated confidence interval can potentially be much smaller than that obtained with SRS measurements, thus reflecting decreased uncertainty.

This encouraging feature results because measurements obtained through RSS are likely to be more regularly spaced than those obtained through SRS and therefore are more representative of the population. The RSS procedure induces stratification of the whole population at the sample level; in effect, we are randomly sampling from the subpopulations of predominantly short, medium, and tall students without having to construct the subpopulation strata.

How is Ranked Set Sampling Applied at a Waste Site?

As mentioned previously, to create ranked sets we must partition the selected first phase sample into sets of equal size. In order to plan an RSS design, we must therefore choose a set size which is typically small, around 3 or 4, to minimize ranking error. Let's arbitrarily call this set size “m,” where “m” is the number of sample units allocated into each set. Proceed as follows:

1. Randomly select m^2 sample units from the population.

2. Allocate the m^2 selected units as randomly as possible into m sets, each of size m.

3. Without yet knowing specific values for the constituent of interest, rank the units within each set based on indicator values for this constituent. This may be based on field screening or done with measurements of a covariate which is correlated with the variable of interest.

4. Choose samples for definitive analysis by including the smallest ranked unit in the first set, then the second smallest ranked unit in the second set, continuing in this fashion until the largest ranked unit is selected in the last set.

5. Repeat steps 1 through 4 for “r” cycles until the desired sample size is obtained for analysis. The sample size is determined by the calculation in Section 7.9.3.

As an illustration, consider the set size m=3 with r=4 cycles. This situation is illustrated below where each row denotes an ordered set of samples within a cycle (S = sample selected for ranking but not selected for definitive analysis, and the units selected for definitive analysis are designated by the letter "X").

In each cycle three sets of three samples each are selected and ranked. In each cycle one sample from each set is selected for analysis; low from the first set, medium from the second set, and high from the third set.. Note that 36 units have been randomly selected in 4 cycles; however, only 12 units are actually analyzed to obtain the ranked set sample of measurements.

Low	**Med.**	**High**	
X	S	S	
S	X	S	1st cycle
S	S	X	
X	S	S	
S	X	S	2nd cycle
S	S	X	
X	S	S	
S	X	S	3rd cycle
S	S	X	
X	S	S	
S	X	S	4th cycle
S	S	X	

Continue selecting sample sets until enough cycles have been completed that the sample number is equal to or greater than the required sample number. For instance, if 47 samples were required you would select samples for six cycles (54 samples, 18 for analysis) so that an equal number of low, medium, and high samples is sent for definitive analysis.

Obtaining a sample in this manner results in maintaining the unbiased nature of simple random sampling. By incorporating "outside" information about the sample units, we are able to contribute a structure to the sample that increases its representativeness of the true underlying population.

If we quantified the same number of sample units, mr = 12, by a simple random sample, we have no control over which units enter the sample. Perhaps all the 12 units would come from the lower end of the range, or perhaps most would be clustered at the low end while one or two units would come from the middle or upper range. With simple random sampling, the only way to increase the prospect of covering the full range of possible values is to increase the sample size. With ranked set sampling, however, we increase the representativeness with a fixed number of sample units, thus saving considerably on quantification costs.

With the ranked set sample thus obtained, unbiased estimators of several important population parameters can be calculated, including the mean and, in the case of more than one sampling cycle, the variance.

Ranking Criteria

The real key to success lies in the ranking procedure. A hazardous waste site inspector may be able to reliably rank areas of soil with respect to concentrations of a toxic constituent, based on field screening methods or other low cost tests, for example a PCB field test kit.

On the other hand, if another characteristic is available that is highly correlated with the characteristic of interest but costs much less to obtain, then we may rank by the values of such a “covariate”. For example, measurement of total organic halides (TOX) in soil in order to rank soil sampling units with respect to the concentration of volatile organic solvents. As an indicator variable, TOX is much less expensive to measure than specific organic compounds.

7.9.5 Sample Collection and Analysis

This section discusses the dynamic workplan and adaptive sampling process, adaptive sampling and analysis strategy, field measurements for constituents, representativeness, and measurement accuracy.

7.9.5.1 The Dynamic Workplan and Adaptive Sampling Process

In the traditional approach, major decisions concerning the direction of the site investigation or cleanup are generally made by the project manager after the field work has been completed. Typically, several field mobilizations occur, reports are written, and many meetings are held between the site owner, its environmental consulting company, and regulatory agencies. In contrast, in an adaptive sampling and analysis program many of these same decisions are made in the field.

In constructing the dynamic workplan, it is important to determine prior to mobilization what decisions will be made, how these decisions will be made, and who will make them in the field. *To assure efficient, effective decision-making IDEM must be included in the development of the dynamic workplan. IDEM must approve all decisions related to sampling and closure.*

Activity 1: Select the core technical team whose responsibility it will be to prepare the dynamic workplan.

The technical team should possess expertise in analytical chemistry, geology, geochemistry, geophysics, hydrogeology, and risk analysis. The team helps with data management, QA/QC, risk assessment, fate and transport modeling, remedial action, community relations, and health and safety. This team will be composed of a mixture of site owner employees and IDEM staff. The technical team will be responsible for the following:

1. Gathering all available information for the site
2. Developing an initial "conceptual" model for the site
3. Identifying the technical objectives and goals to be accomplished
4. Supervising the field effort, making adjustments to the CSM based on the data produced in the field
5. Evaluating the conceptual model and decisions made with respect to federal, state, and local regulations

This core technical team will be responsible for making decisions in the field. One member of the team must have final decision making authority and responsibility. This helps keep the site investigation process moving forward at a reasonable pace. At least one member of the technical team should be on site at all times. This technical team member and an IDEM staff member must be on site when sampling activities take place. These people must have a working knowledge of all aspects of the investigation and cleanup DQOs, and must routinely communicate with other technical team members.

Field personnel (and off-site technical team members) should be in regular communication with appropriate staff from IDEM to ensure that decisions made in the field, typically under the pressures of time and field-resource utilization, are in conformance with the dynamic workplan framework and any other requirements placed on the site investigation.

Activities 2 through 6 are often considered to be part of the Quality Assurance Project Plan (QAPP).

Activity 2: Develop the Initial Conceptual Model and Decision Making Framework.

The CSM. The initial conceptual model contains the best information available at the start of the project. It depicts the three-dimensional site profile based on vadose zone and ground water flow systems that can exert influence on constituent movement. Key site features such as roads, buildings, hydrography, depth to bedrock, direction of ground water flow, and potential preferential pathways for constituent transport are mapped. Map cross sections should include water levels, high and low permeability zones, and aquifers. Chapter 2 and the RISC software provide a framework for developing the CSM.

The CSM is updated as additional data becomes available during the site investigation and cleanup process. It is the basis for the dynamic workplan. The CSM changes to reflect the increased site knowledge gained from field activities.

Stakeholders should (1) agree at the beginning on the most likely kinds of actions to be taken as a result of the field data, (2) implement the appropriate action on a daily basis as the data is generated, and (3) take new directions when the data suggests deviations from the conceptual model.

Site delineation is an iterative process and should be viewed as an ongoing experimental project.

The Decision Making Framework. The initial conceptual model is based on the DQO for the site. The DQO process involves a series of planning steps designed to ensure that the type, quantity, and quality of environmental data used in decision making are appropriate for the intended application. It relates data needs to specific decisions to be made.

See Appendix 6 and Chapter 6 for more information on developing DQOs.

Activity 3: Develop Standard Operating Procedures

The next step in developing a dynamic workplan is to establish standard operating procedures (SOP). SOPs for sample collection and analysis should be produced along with other SOPs required to answer site-specific questions, such as geophysical and hydrogeological surveys. The SOPs should be developed by the site owner's core technical team and approved by IDEM prior to initiating field activities.

Field methods should be performance based and provide data of sufficient quality to meet the DQOs. Because these technologies and methods may not be amenable to typical CLP or SW846 methods, QC procedures or data reporting formats, supporting data produced from the proposed field techniques should be provided to document data quality. *Note: While not always required for field data, CLP and SW-846 methods (as appropriate) are always required for laboratory samples.*

Activity 4: Develop the Data Management Plan

Critical to the success of the dynamic process is the ability to manage and easily use all of the data produced in the field. Data integration (chemical, physical, geological, hydrological), sampling, and analysis protocols should be incorporated into an overall data management plan. Protocols for sample logging, analysis, data reduction, and site mapping should be established. Several different organizations may be involved in this process. The data management plan should be established with rules and responsibilities defined prior to mobilization for the collection, assimilation, and presentation of the field generated data. As an example, computers housed in the laboratories can be electronically linked to the data management trailer on site.

Sample logging information and the results of the analyses can be managed through a Laboratory Information Management System or through the use of spread sheets. The data can then be downloaded to a computer containing site visualization software for conceptual model update and review. This easy access to analytical and site information simplifies the on-site decision making process.

Activity 5: Develop the Quality Assurance Project Plan

This document contains the sampling methods, analytical procedures, and appropriate quality assurance (QA) and quality control (QC) procedures. It describes the procedures to be used to monitor conformance with, or justification for departure from the SOPs. The overall goal is to ensure that data of known and adequate quality have been produced to support the decision making process.

Activity 6: Prepare the Health and Safety Plan

Finally, a health and safety plan is produced as part of the Dynamic Workplan/Adaptive Sampling and Analysis project. Procedures must be established for safe use of the field analytical tools and for the methods used to monitor worker and community safety.

7.9.5.2 Adaptive Sampling and Analysis Strategy

The number of sampling rounds made during a field mobilization is dependent on the DQO specifications for confirming the presence or absence of constituents. Once the soil contamination profile objectives have been met and a verified conceptual model is produced, the data should be capable of identifying which of the two categories a particular source area falls within:

- The site is clean or poses negligible risk, and no further action is required.
- The site is contaminated at concentrations that exceed action levels for negligible risk; remedial action or other measures are required.

For those constituents found in the first round of sampling, target compound analysis is performed in each subsequent sampling round. As the analyte list decreases, more samples for each specific constituent may be analyzed during the workday.

If site samples contain no detectable constituents above the closure levels established for the site, closure sampling may be done. Closure sampling is always done by random sample design with off-site laboratory analysis.

If site screening measurements result in COC concentrations greater than the closure levels, sampling continues and the conceptual model is refined until the site-specific DQOs are met. Once the site data and conceptual model are verified, risk-based decision making occurs with respect to human health and the environment. At this point, new workplans must be produced to address site remediation needs.

Sampling may be directed by geostatistical sampling tools which are able to predict where the next round of samples should be collected. Because quantitative measurements are made on-site, greater confidence should be obtained in the sampling. If screening quality data, such as enzyme kits are used for initial sampling, quantitative analytical data should be produced to verify the results from the site screening phase.

The number of locations within and surrounding each contaminated and non contaminated area as well as the depth of samples at each location should be determined by the core technical team. In an adaptive sampling and analysis program, contaminated areas are more heavily sampled than in traditional site characterization studies. Therefore, if semiquantitative or quantitative field analytics is performed, only 10 to 25 percent of the samples will need laboratory verification. The percentage depends on the specifications of the method used. These samples should be selected in a random manner.

Field results will differ from off-site laboratory results for volatile organic compound (VOC) contaminated soil samples, with field measurements generally producing higher measurement concentrations because of analyte loss during off-site sample transport and storage. Care must be taken when these types of comparisons are made. Because site investigation and cleanup decisions are made based on field data, off-site laboratory analysis should be restricted to about 10 percent of the samples analyzed when a quantitative field laboratory.

As additional data is obtained it will help refine the conceptual model and dictate future directions. Site work stops when answers to the questions posed in the workplan meet site-specific confidence levels established as part of the DQO process. To ensure that site-specific goals have been met, the project team should statistically evaluate the results of its findings. An adaptive sampling and analysis program focuses staff, equipment, and financial resources in areas where contamination exists, while providing a more limited evaluation in areas that pose little risk to human health and the environment.

7.9.5.3 Field Measurement for Chemicals of Concerns

The selection of field analytical methods is based on the need to make quick decisions in the field. Field analytical techniques should be capable of providing data in a matter of minutes. They should have documented measurement sensitivity, precision, and accuracy so that instruments can be matched with site investigation and cleanup DQOs.

The simpler the technique, the more likely it will be used in the field. Field instruments must be transportable, operate under adverse conditions, and provide improved cost/benefit over laboratory analysis. For projects of short duration and low sample volume, staff and equipment mobilization expenses may make field analytics a cost-prohibitive option. In addition, if quantitative measurements are required for all samples, field analytics may not provide a cost-effective means for obtaining site data.

The selected field method must demonstrate method detection limits at approximately half the cleanup level established for the site. Using field methods of this accuracy site decisions can be made, including:

- The Nature and Extent of Contamination – Field data supports the overall site investigation
- Risk to human health and the environment – Field data provides input into the risk assessment process
- Achievement of cleanup objectives – Field data supports site compliance with acceptable constituent levels

To insure that the field analytical instrumentation and methods selected in the workplan are amenable to a given site, site-specific method detection limit studies should be performed for each class of COCs (for example, VOCs, semivolatile organic compounds, and metals) using soil obtained from the site prior to the field investigation. This will help to determine whether matrix interferents or target compounds mask (for example, portable gas chromatograph [GC]) or cross-react (for example, enzyme/wet chemical kits) with targeted organics or metals (for example, by electrochemical detection).

7.9.5.4 Representativeness

Representativeness is the degree to which data accurately and precisely represents the frequency distribution of a specific variable. Measurement accuracy can be influenced by measurement sensitivity, selectivity, and precision whereas representativeness is affected by sampling location and sampling methods. The influence of sampling on analytical quality is extremely significant.

Sample values have little meaning unless they are representative of concentrations across the site. The following factors may affect sample representativeness:

- Geological Variability — Regional and local variability in the mineralogy of rocks and soils, the buffering capacity of soils, lithogic permeability, and variability in the sorptive capacity of the vadose zone

- COC Concentration Variability — Variations in the COC concentrations throughout the site

- Collection and Preparation Variability — Deviations in analytical results attributable to bias introduced during sample collection, preparation, and transportation

- Analytical Variability — Deviations in analytical results attributable to the manner in which the sample was stored, prepared, and analyzed by the on-site or off-site laboratory. Although analytical variability cannot be corrected through representative sampling, it can falsely lead to the conclusion that error is due to sample collection and handling procedures.

The variability in soil COC concentrations often makes it too costly to use traditional site investigative approaches because it may be difficult to collect the number of samples needed to have confidence that the extent, direction, concentration, and rate of COC movement have been correctly delineated. The adaptive sampling and analysis strategy helps to focus the intensive sampling efforts on areas where contamination has been identified, producing more data in the areas where it is needed.

7.9.5.5 Measurement Accuracy

Assuming representative samples have been collected, measurement accuracy is directly dependent on the relationship among three key analytical parameters: *precision, selectivity, and sensitivity*. Accurate results cannot be obtained unless the measurement technique produces selective detection and adequate sensitivity. Selectivity refers to the instrument’s or method’s ability to respond to target compounds in the presence of nontarget sample constituents.

For example, if the analytical technique responds to the presence of matrix interferents or cross-reactive target compounds, measurement identity is affected and thus, accuracy. Moreover, if the analyte concentrations in the sample are at or just below the method detection limit, the measured concentrations may show poor precision due to lack of sensitivity.

Measurement precision is the degree to which a set of analyses of the same parameter are repeatable. To achieve unambiguous analyte identification and the desired method detection limit, extensive sample preparation procedures may be required to remove matrix constituents, dilute, or pre-concentrate the sample extract. These additional steps lengthen the overall time of the analysis, reducing the sample throughput rate.

Generally, as one property of the equilateral triangle is improved, one or both of the remaining analytical properties can become distorted. For example, increasing the number of sample preparation steps prior to the analytical measurement can result in loss of analyte, which, in turn, can influence measurement sensitivity and thus, accuracy (false negative). Another example is the detection of nitrated explosives by selective reagents such as enzymes. Field-practical enzyme immunoassay kits can significantly reduce the time of analysis over laboratory high performance liquid chromatography (HPLC) methods by eliminating the need for sample cleanup procedures. False positive detection is possible, however, due to cross-reactivity with other nitrated organic compounds that might be present in the sample.

Although advancements in analytical methods have increased laboratory productivity, sample throughput rates and data quality are greatly influenced by interactions among selectivity, sensitivity, and precision. As increasingly more stringent measurement accuracy is specified, sample throughput rates decrease.

7.9.6 Quality Assessment

When the SRS or RSS has been collected, the samples are sent to the laboratory for analysis. Be sure that all QA/QC procedures appropriate to the desired sample quality are followed. The report from the laboratory should contain all information needed to perform data validation procedures.

Risk assessment and site management work relies heavily on statistics. There are five basic activities performed by the statistician during the data quality assessment process.

1. Review data quality objectives to ensure that appropriate environmental decision criteria are used, to define the statistical hypothesis, to specify tolerable limits for decision errors, and to define acceptable confidence limits or probability interval width.

2. Perform a preliminary data review which includes: a review of QA reports to ensure that data quality is appropriate, calculation of basic statistics (mean, standard deviation, range, and others), and generation of data graphs. This information is used to learn about the structure of the data and identify patterns, relationships, or potential anomalies.

3. Select the most appropriate procedure for summarizing and analyzing the data based on information gathered in activities 1 and 2. This includes identifying the underlying assumptions that must hold for the statistical procedures to be valid.

4. Verify the assumptions of the statistical tests. This includes an evaluation of whether the underlying assumptions hold, or whether departures are acceptable given the actual data and other available information.

5. Perform the calculations required for the statistical tests and draw conclusions from the data. Document the inferences drawn as a result of the calculations. If the design is to be used again, evaluate of the performance of the sampling design.

Data collection and laboratory analysis provide estimates of the environmental concentration of constituents. Statistics give assurance that the estimates are accurate within established limits.

7.9.7 Additional Information

This section provides additional references on dynamic workplans, soil sampling, and ranked set sampling.

Dynamic Workplan References

Robbat, A. 1998. “Dynamic Workplans and Field Analytics: The Keys to Cost-Effective Site Characterization and Cleanup” Written for EPA

EPA. 1991. “Removal Program Representative Sampling Guidance” OSWER Directive 9360.4-10

General Soil Sampling References

Barth, D. S., Mason, B. J., Starks, T. H., and K. W. Brown. 1989. Soil Sampling Quality Assurance User's Guide, Second Edition, EPA Cooperative Agreement No. CR 814701, Environmental Monitoring Systems Laboratory, Las Vegas, Nevada.

Bross, I. D. 1985. "Why Proof of Safety is Much More Difficult Than Proof of Hazard." Biometrics. Vol. 41: 785-793.

EPA. 1983. Preparation of Soil Sampling Protocol: Techniques and Strategies, Washington D.C., August 1983 (EPA 600/4-83-020).

EPA. 1984. Soil Sampling Quality Assurance User's Guide, First Edition, Washington, D.C., May 1984 (EPA 600/4-84-043).

EPA. 1985. Verification of PCB Spill Cleanup by Sampling and Analysis, Washington D.C., August 1985 (EPA 560/5-85-026).

EPA. 1986d. Superfund Public Health Evaluation Manual, Washington D.C.: EPA [October 1986].

EPA. 1987a. Data Quality Objectives For Remedial Response Activities: Development Process, Washington D.C., March 1987 (EPA 540/G-87/003).

EPA. 1987b. Data Quality Objectives For Remedial Response Activities: Example Senario RI/FS Activities at a Site with Contaminated Soils and Ground Water, Washington D.C., March 1987 (EPA 540/G-87/004).

EPA Center For Environmental Statistics, “Soils and Solid Media”

Finney, D. J. 1948. "Random and Systematic Sampling in Timber Surveys." Forestry. Vol. 22: 64-99.

Ford, P., and P. Turina. 1985. Characterization of Hazardous Waste Sites--A Methods Manual, Volume I--Site Investigations. Las Vegas, Nevada: EPA Environmental Monitoring Systems Laboratory, [April 1985].

Gilbert, R. O. 1982. Tran-Stat: Statistics for Environmental Studies, No. 19, Some Statistical Aspects of Finding Hot Spots and Buried Radioactivity. Richland, Wash.: Pacific Northwest Laboratory.

Gilbert, R. O. 1983. Tran-Stat: Statistics for Environmental Studies, No. 24, Field Sampling Designs, Simple Random and Stratified Random Sampling. Richland, Wash.: Battelle Pacific Northwest Labs.

Gilbert, R. O. 984. Tran-Stat: Statistics for Environmental Studies, No. 26, Field Sampling Designs: Systematic Sampling. Richland, Wa.: Battelle Memorial Institute, Pacific Northwest Laboratory.

Gilbert, R. O. 1987. Statistical Methods for Environmental Pollution Monitoring. New York: Van Nostrand Reinhold.

Gilbert, R. O., and P. G. Doctor. 1985. "Determining the Number and Size of Soil Aliquots for Assessing Particulate Contaminant Concentrations." Journal of Environmental Quality. Vol. 14: 286-292.

Gilbert, R. O., and R. R. Kinnison. 1981. "Statistical Methods for Estimating the Mean and Variance from Radionuclide Data Sets Containing Negative, Unreported or Less-Than Values." Health Physics. Vol. 40: 377-390.

Gilbert, R. O., Miller, M. L, and H. R. Meyer. 1989. "On the Design of a Sampling Plan to Verify Compliance with EPA Standards for Radium-226 in Soil at Uranium Mill Tailings Remedial-Action Sites." ASA/EPA Conference Proceedings on Interpretation of Environmental Data. IV Compliance Sampling, October 5-6, 1987 (EPA-230-03-047).

Palley, M. N., and L. G. Horwitz. 1961. "Properties of Some Random and Systematic Point Sampling Estimators." Forest Science. Vol. 77: 52-65.

Parkhurst, D. F. 1984. "Optimal Sampling Geometry for Hazardous Waste Sites." Environmental Science and Technology. Vol. 18, No. 7: 521-523.

Peshkova, N. V. 1970. "Comparison of Results of Determinations of the Density of Plant Populations by Systematic and by Restricted Random Sampling Procedures." Ecology (Ekologiya). Vol. 2: 180-181.

Singer, D. A. 1972. "Elipgrid. A Fortran IV Program for Calculating the Probability of Success in Locating Elliptical Targets With Square, Rectangular, and Hexagonal Grids." Geological Programs. Vol. 4: 1-16.

Singer, D. A. 1975. Relative Efficiencies of Square and Triangular Grids in the Search for Elliptically Shaped Resource Targets. Jour Research U. S. Geological Survey. Vol. 3, No. 2: 163-167.

Wolter, K. M. 1984. "Investigation of Some Estimators of Variance for Systematic Sampling." Journal of the American Statistical Association. Vol. 79, No. 388: 781-790.

Zirschky, J., and R. O. Gilbert. 1984. "Detecting Hot Spots at Hazardous Waste Sites." Chemical Engineering. Vol. 91: 97-100.

Ranked Set Sampling References

U.S. EPA Observational Economy Series, Ranked Set Sampling, M.T. Boswell, S.D. Gore, G.D. Johnson, G.P. Patil, C. Taillie, Penn State University Center for Statistical Ecology and Environmental Statistics; John Fritzvold, Herbert Lacayo, Robert O'Brien, Brenda Odom, Barry Nussbaum, John Warren, U.S. EPA Project Officers.

Kaur, A., Patil, G. P., and Taillie, C. (1994). Unequal allocation model for ranked set sampling with skew distributions. Technical Report 94-0930, Center for Statistical Ecology and Environmental Statistics, Department of Statistics, Pennsylvania State University, University Park, PA.

Kaur, A., Patil, G. P., Sinha, A. K. and Taillie, C. (1995). Ranked set sampling: An annotated bibliography. Environmental and Ecological Statistics, 2(1) (to appear).

Patil, G. P., Sinha, A. K., and Taillie, C. (1994). Ranked set sampling. In Handbook of Statistics, Volume 12: Environmental Statistics, G. P. Patil and C. R. Rao, eds. North Holland/Elsevier Science Publishers.

Stokes, S. L. (1980a). Estimation of variance using judgment ordered ranked set samples. Biometrics, 36, 35-42.